Journal of Hydrology 573 (2019) 676–687

Contents lists available at ScienceDirect

Journal of Hydrology

journal homepage: www.elsevier.com/locate/jhydrol

Research papers

Modelling the impact of climate change on flow and *E. coli* concentration in the catchment of an ungauged drinking water source in Norway

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ARTICLE INFO

This manuscript was handled by Marco Borga, Editor-in-Chief, with the assistance of Patrick Norman Lane, Associate Editor

Keywords:
Hydrological model
Climate change
Streamflow
Flow patterns
E. coli
Drinking water source

ABSTRACT

The discharge of fecal indicator bacteria into surface water sources poses treatment challenges to drinking water utilities. Changes in the discharge regimes of streams resulting from extreme precipitation associated with climate change can exacerbate these challenges. The catchments of many surface water sources depended upon by water treatment plants are ungauged, thus making it difficult to predict the impacts of climate change on the microbial quality of such water sources. In this study, a hydrological model is implemented to assess the impact of local climate projections on flow regimes and *E. coli* discharge in the ungauged catchment of the drinking water source for the city of Ålesund in Norway. We performed separate sensitivity analysis for flow and *E. coli,* and assessed the variations in patterns of flow and *E. coli* concentrations in the streams in future using adjusted meteorological data. The calibrated flow showed good agreement with the observed data, but the *E. coli* model yielded lower performance indices. The results indicate that interception of precipitation by forest cover and soil organic carbon content mostly control flow generation in the catchments, and the variations in the concentrations of *E. coli* were more sensitive to instream sediment processes. Flow levels in the streams would remain the same in 2045 and 2075. Peak flows that currently occur in spring and autumn months may respectively shift back to winter and summer months. High concentrations of *E. coli* currently observed in summer in the streams may be lower in future, but the concentrations may be high at the start of autumn seasons. The results have management implications for watersheds and water utilities mostly in temperate climate regions.

1. Introduction

Climate and weather conditions practically affect every aspect of the environment, and the intensities of primary weather variables such as precipitation and temperature are likely to increase in the 21st century due to climate change (IPCC, 2014). The Norwegian climate has become warmer, and precipitation has increased by 20% over the past 100 years. Current climate projections in Norway indicate that precipitation may rise by 5–30%, while temperature is expected to increase by between 2.3 °C and 4.6 °C (Norwegian Ministry of Climate and Environment, 2014). These projected changes in climatic variables are expected to impact on the quality and quantity of surface water sources that account for more than 90% of the country's drinking water supply (Bates et al., 2008; Brown et al., 2013; Norwegian Ministry of Climate and Environment, 2014). Climate change related variations in air temperature and hydrological regimes of surface water sources can significantly cause deteriorations in the quality of water. Variations in precipitation may increase or decrease the concentrations of microorganisms in receiving surface waterbodies through increased runoffs, sediment transport and resuspension in streams (Hofstra, 2011; Aragonés et al., 2016; Lucas et al., 2014).

Observed trends in the deterioration of drinking water quality under changing climate can increase public health risk (Delpla et al, 2009). In addition, climate change is expected to vary the availability, variability and seasonality of discharge in rivers (Whitehead et al., 2015), affect the incidence of waterborne pathogens, and pose greater management problems in the future (Patz and Hahn, 2013). Accordingly, there is a growing need for investigating potential changes in hydrological extremes of surface waterbodies (Stahl et al, 2012) and predicting faecal indicator bacteria (FIB) concentrations, fluxes, and loads in a variety of waterbodies to provide the necessary policy framework and guidelines to address water quality challenges in the future (Cho et al., 2016). The severity of climate impacts on the quality of surface water bodies does not only depend on the magnitude of the expected changes in climate

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https://doi.org/10.1016/j.jhydrol.2019.04.021
Received 12 April 2018; Received in revised form 2 April 2019; Accepted 5 April 2019
Available online 05 April 2019
0022-1694/ © 2019 Published by Elsevier B.V.

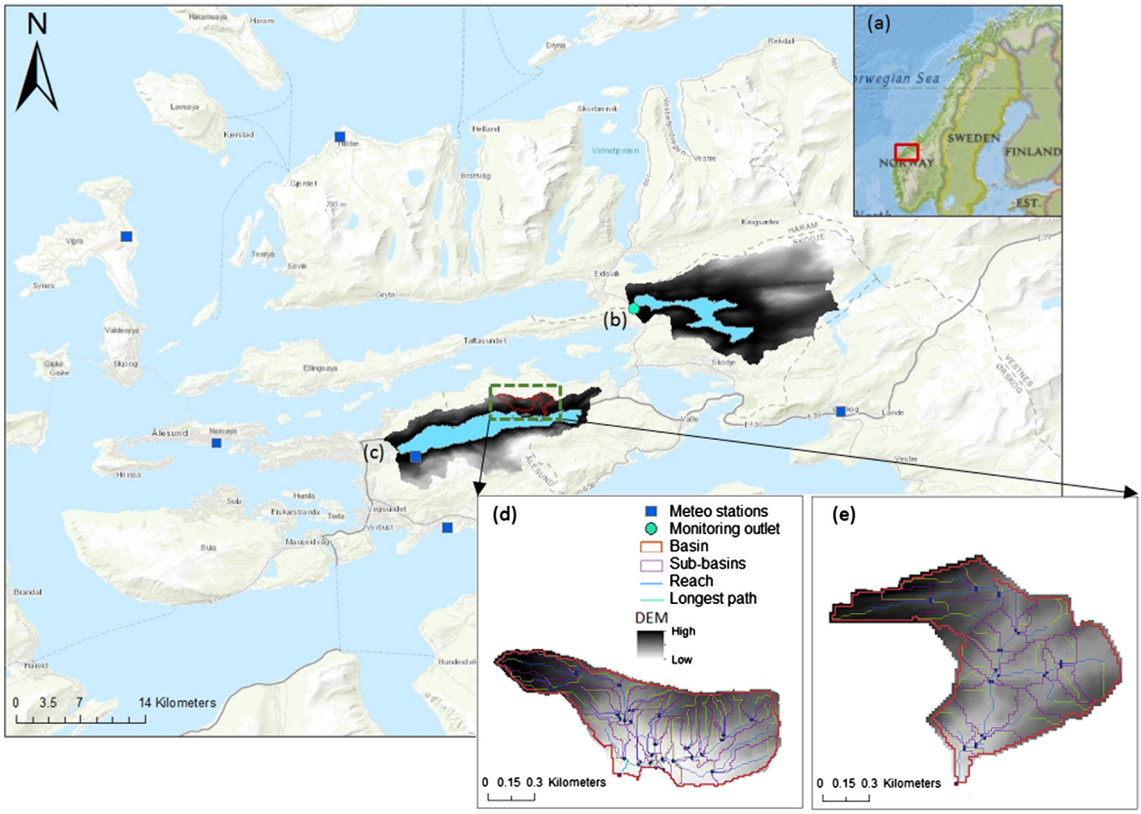

Fig. 1. Study area: (a) location of study site in Norway; (b) guaged catchment (Engestdalvatnet Lake); (c) ungauged drinking water source catchment (Brusdalsvatnet Lake); (d) Slettebakk stream sub-catchment; (e) Brusdalen stream sub-catchment.

variables, but also developments in future land use. In addition, the negative impacts of climate change on the quality of raw water sources can be reduced if water utilities have the capacities to plan and implement effective adaptation strategies within the catchments of their raw water sources that take into account the effects of climate change. The development of such strategies require an integrated approach that links climate change projections to hydrological processes within the catchments of raw water sources. For water utilities that mainly rely on surface water sources, it is essential to evaluate and understand the imminent management challenges and potential health risks that may be associated with variabilities in the pattern of microbial discharge in the future due to climate change. Securing long-term safety of the drinking water source is necessary for sustainable management of drinking water supply systems.

Investigating the impacts of climate change on hydrological (flow) and water quality regimes of raw water sources usually involve the application of hydrological models using historical climate data and projected data obtained by adjusting historical observations based on assumed future climate scenarios (Wilby and Harris, 2006). For this, a number of studies have examined the effects of climate change on flow regimes at catchment scales (Graham et al., 2007; Prudhomme & Davies, 2009; Vansteenkiste et al., 2013; Meresa et al., 2017). In the work of Meresa et al. (2017), in which the trends in selected hydro-climatic indices in Norwegian and Polish catchments were investigated, the authors observed that periods of increased precipitation in catchments with rainfall-driven floods were followed by increased flow levels, with strong seasonal changes. For catchments with snow-driven flood regimes, however, decreases in annual maximum flows were observed over the studied period (Meresa et al., 2017). Moreover, results of a study that investigated hydrological regimes under future climatic conditions in central Norway revealed that climate change would strongly influence stream flow, with more evenly distributed flow volumes across seasons, and significant reduction in the amount of snow cover (Hiriyanna Rao Ravindra, 2017). In respect of modeling the discharge of FIB into surface water, the focus of many studies has been on improving the understanding of the effects of hydrological processes on their fate and transport in the environment (Jonsson and Agerberg, 2015; Niazi et al., 2015), as well as the effects of in-stream processes (Kim et al., 2017). To date, few studies have linked the discharge of faecal indicator organisms FIB to climate-induced changes to flow patterns that may occur in future (Liu and Chan, 2015). To our knowledge, no study has been undertaken in Scandinavia to account for the effects of climate change on the hydrological response and water quality (*E. coli* concentrations) within the catchments of drinking water sources.

Further, although water supply systems in Norway have improved over the last 20–30 years, most of the catchments of drinking water sources are ungauged. The ungauged nature of the catchments inhibits the development of effective strategies for water safety planning, especially on the impact of climate change on hydrological processes and water quality within the catchments. There are existing approaches involving the transfer of hydrological parameters in catchments that share common spatial characteristics to help address this challenge. In this study, hydrological model using existing approaches is implemented to assess the impact of local climate projections on flow regimes and E. coli discharge in the ungauged catchment of the drinking water source for the city of Ålesund in Norway. The approach used in this study is expected to be replicated for other ungauged catchments in Norway, particularly those serving drinking water treatment plants; and are potentially vulnerable to climate change. In addition, the results of the study are expected to contribute to improvement in understanding of potential impact of climate change on the quality of surface water drinking water sources in Norway.

2. Materials and methods

2.1. Brusdalsvatnet Lake and catchment

The streams modelled in this study discharge into the Brusdalsvatnet Lake, which is located within the inland of Oksenøya in the Møre and Romsdal County between Ålesund and Skodje municipalities in the west coast of Norway (Fig. 1). The West Coast of Norway is a mountainous and densely forested region that receives significant amount of rainfall throughout the four seasons of the year, though peak rainfalls occur mostly in spring (March – May) and autumn (September – November). Brusdalsvatnet Lake provides drinking water to the 50,000 inhabitants of the city of Ålesund and neighbouring Sula municipality.

The Lake has a surface area of approximately 7.21 km^2, with a catchment area of approximately 28.1 km^2. The Lake has a mean surface elevation of 26 m above sea level. The altitude of the catchment therefore varies from 26 m (Lake surface) to 559 m in the southern and southeastern sections of the catchment. The mean elevation in the northern side of the lake is approximately 200 m. The deepest section of the Lake is 94 m, located at the central part of its longitudinal section. The drinking water treatment plant draws 55,000 cubic meters of water daily at a depth of 35 m from the southwestern section of the Lake. The northwestern and southwestern sections of the Lake catchment are sparsely populated with settlements, while the other sections are uninhabited, with little anthropogenic activities.

Among the tributaries of the Lake are two rivers; Arsetelva and Vasstrandelva, which are located on the southern side, and two major streams; Slettebakk and Brusdalen (Fig. 1c). Results from an ongoing water quality survey in these tributaries show that the two streams; Slettebakk and Brusdalen discharge very high concentrations of *E. coli* (551–1160 CFU/100 ml) into the Lake, compared to average concentrations of 11 CFU/100 ml observed in the other major tributaries of the Lake. Thus this study focused on modeling water flow and *E. coli* discharges from these streams (Fig. 1d & e). The gauged catchment (Engesetdalsvatnet Lake) from which discharge and hydrological model parameters were transferred to the target catchment is shown in Fig. 1b. Comparison of the spatial and meteorological characteristics of the donor and target catchments are further presented in Table 1.

2.2. Regionalization of flow data from gauged to ungauged catchment

The target catchment in this study (Brusdalsvatnet Lake) is currently not equipped with gauging stations. Accordingly, obtaining estimates of the discharge into the streams in this catchment required the use of regionalization. Usually, catchments that are within the same climate region and with similar characteristics such as topography, soil type, land use, precipitation, and temperature have similar hydrological behavior. Hydrological model parameters for such ungauged catchments can be estimated by regionalization (Bárdossy, 2007). This involves transferring the hydrological parameters of the gauged catchment onto the ungauged. Methods applied in such transformations include area ratio transformation, regression, (Bastola et al., 2008), physical similarity (McIntyre et al., 2005), spatial proximity (Merz and Blöshla, 2004), or more advanced methods such as kriging (Parajka et al, 2005) and those based on satellite radar altimetry observations (Sun et al., 2012). In this study, we applied the area ratio method, to transfer measured flow data from the donor catchment to our target catchment. The hydrological model was first calibrated for the donor catchment, and the parameters applied to the target. Finally, the transferred flow was used for comparison with the estimated flows in the target streams. The characteristics of the donor and target catchments in this study are shown in Table 1. In this table, rainfall and runoff are presented in units of average depth of water over the drainage basins per year (mm/year). It can be seen in Table 1 that runoff in the donor catchment (1732 mm/year) is higher than the annual mean rainfall in the catchment (1704 mm/year). Except for the characteristics of precipitation in the catchment such as type (rainfall and snow), intensity, duration, distribution and antecedent precipitation, other factors can influence the amount of runoff generated. These include watershed characteristics such as size, shape (or form factor), altitude, slope, land use and depressions. In addition, the donor catchment is in a higher altitude where storage of snow occurs for a longer period in the year, potentially contributing to the higher runoff.

Table 1
Characteristics of donor and target catchments.

	Donor catchment	Target catchments	
	Engesetdalsvatnet	Slettebakk	Brusdalen
Area (km^2)	40.4	0.9	0.5
Elevation (m) Min.	43	26	29
Max.	747	296	190
Mean	207	134	110
Mean Slope (%)	24.7	31.4	25.2
Mean annual precipitation (mm)	1704	2073	1973
Mean annual temperature (°C)	5.5	6.3	6.3
Channel length (km)	11.8	0.5	1
Forest cover (%)	67.58	95.57	83.33
Annual runoff (mm/yr.)	1732.21	1252.20	1180.48

2.3. Spatial and geomorphological characteristics of catchment

Spatial data needed to specify the catchment attributes included land use and soil types. Digital maps for land use was obtained from the Forest and Landscape Services unit of the Norwegian Institute for Bioeconomy (NIBIO), while the Norwegian Geological Survey (NGU) provided data for soil and sediment types in the catchments. It can be seen from Table 1 that the annual estimates of the precipitation in the donor catchment is significantly lower (approximately 15%) than in the target catchments, despite the relatively larger size of the former. Relative to the donor catchments, the target catchment is located closer to the coast. The coastal areas in the region generally receive higher rainfall than the inland areas, possibly due to interception of the clouds by the mountains in the area. In addition, the donor catchment is composed of approximately 67% forest, while the Slettebakk and Brusdalen catchments (target) are respectively comprise of 95% and 84% forest cover. This relatively larger proportion of forest canopies in the target catchments can substantially favor the amount of precipitation in the catchment. Despite the relatively larger size of the donor catchment, the differences in the annual runoff is approximately 38% compared to the target catchment. This may be caused by differences in other watershed characteristics as mentioned in Section 2.2 of this study. Subsequently, some differences in the parameters that control runoff generation in the two catchments may be expected in the hydrological model, potentially affecting the model predictions. For this reason, additional calibration was performed for the target catchment for use in predicting the future runoff as explained in Section 2.7.4 of this study.

As shown in Table 1, the Slettebakk and Brusdalen streams modelled in this study are relatively small, with channel lengths of 0.5 Km and 1 Km respectively. Due to the steep topography of the catchment, the widths of the channels vary significantly, and the widest sections of the two streams (Slettebakk and Brusdalen) are approximately 1 m and 1.5 m respectively for. In addition, the base flow in both streams is 0.01 l/s. Although there are no specific data on sediment size distributions in the streams, average concentrations of suspended sediments in streams located in the study region (west coast of the Norway) have been reported to range from 13 mg/l to 24 mg/l (Bechmann et al., 2008).

While no specific farming activity exists in the Slettebakk stream catchment, farm animals including sheep and horses are located in the

Brusdalen sub-catchment. Similarly, the dominant soil types in the catchments are podzols (spodosols) and moraine, both of which are characterized by high humus content and/or peat cover and are freely drained. The proportions in the donor catchment are; podzols cover (53%) and moraine soil, rocks and sediments (46%), while target catchments of the streams are both composed of 100% podzols.

2.4. E. coli input and calibration data

Biweekly, water sampling and *E. coli* analysis were carried out from the two streams (Slettebakk and Brusdalen) in two periods in 2017, from March to May 2017, and from mid-August to December 2017. A total of 14 samples were collected in sterile 250 ml bottles. *E. coli* analysis was performed using Membrane Filtration (MF) method. The MF method involved filtration of 100 ml of water samples through sterile, single packed, microporous membranes (47 mm diameter, 0.45 ± 0.02 μm pore size) with grid. The membranes were transferred with sterile tweezers on to Coliform Chromogenic Agar (CCA) media (ISO 9308-1, 2000) for *E. coli* cultivation at 37 °C for 24 h. The observed *E. coli* (CFU/100 ml) were used to validate the bacteria model for the two streams in the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998; Arnold et al., 2012). Typical concentrations of *E. coli* from domestic and wild animal manure (horses, sheep and deer that dominate the stream sub-catchments) needed as inputs to the SWAT model were based on values published in a report of the Norwegian Institute for Water Research (NIVA) (Berge et al., 2011). According to the report, average concentrations of *E. coli* per 23 Kg faeces of a horse per day is 5×10^9 CFU and can vary from 10^7 to 10^{11} CFU. The corresponding average value for sheep is 1×10^{10} CFU (with variation of 10^8–10^{11}), and 2×10^8 CFU for deer. There are stables with 10 horses in the Brusdalen stream sub-catchment, but the actual number of sheep in the farm located within the sub-catchment was not ascertained from the farmer. During our regular sampling exercise, we counted 20 sheep in the area close to the stream. Further, visual inspection of the location during sampling indicated that the horse stables and the sheep nearby the stream will contribute to faecal contamination. For the Slettebakk stream sub-catchment, potential population of deer was obtained from the Ålesund local hunting group, who reported 40 individuals seen within 33 days over a wider area, which comprises the sub-catchment. Based on this information and the concentrations published in the NIVA report, the daily loadings of *E. coli* in dry animal manure into the streams were set as 7.5×10^8 CFU/g and 2×10^8 CFU/g respectively for the Brusdalen and Slettebakk sub-catchments.

2.5. Climate data and bias correction

Except for catchment characteristics, hydrological processes in catchments are mainly driven by climatic variables. These include precipitation, air temperature as well as shortwave and longwave solar radiations. In this study, daily precipitation (mm/day), minimum and maximum temperature (°C) were obtained from six weather stations (Fig. 1). The data constituting daily measurements from each of the stations from 2010 to 2017 was taken from an online database provided by the Norwegian Water Resources and Energy Directorate (NVE). The weather stations include Vigra (Lat. 62. 5617, Long. 6.115), Hildre (Lat. 62.6017, Long. 6.3187), Ålesund IV (Lat. 62.4703, Long. 6.2108), Ørskog (Lat. 62.4775, Long. 6.8167), Brusdalsvatn II (Lat. 62.4654, Long. 6.4013) and Emblem (Lat. 62.4317, Long. 6.4325). The elevations of the weather stations ranged from 1 m to 33 m above sea level. Missing data points occurring in records were estimated using linear interpolation. Daily averages were calculated from the six weather stations and used as inputs for the hydrological model. The Soil and Water Analysis Tool (SWAT) program used for the modelling in this study simulated other relevant weather parameters including solar radiation, wind speed, and humidity. In addition, the program calculated potential evapotranspiration using the Penman-Monteith equation.

Table 2
Change factors used for adjusting historical climate data for climate scenarios. The factors were calculated from median projections of precipitation (%) and temperature (°C) over the region (Norwegian Centre for Climate Services).

Year	Winter	Spring	Summer	Autumn
Precipitation				
2045	0.158	14.218	9.789	−13.17
2075	−0.121	17.252	14.263	−10.78
Temperature				
2045	1.922	1.779	1.743	1.975
2075	3.374	3.241	3.112	3.409

To simulate the flows and *E. coli* concentrations in the streams for future scenarios, we used air temperature and precipitation data from the Representative Concentration Pathways (RCP 8.5) climate models for the Møre and Romsdal region of Norway. The RCP8.5 uses three scenarios for emissions of greenhouse gases and estimates climate change for up to 2100. The model uses the period 1971–2000 as the base year and estimates climate change for up to year 2100. Bias correction of the climate data was carried out using change factor method (Arnell, 1996; Hijmans et al., 2005). Mean seasonal outputs of the RCP 8.5 model were taken from the Norwegian Climate Service Center, and the differences between the current values (2010–2017) and future projections (2045 and 2075) were calculated. Thus, to adjust the precipitation data, we calculated the ratios between the RCP 8.5 simulations for future and current climate. The seasonal ratios were multiplied by the historical time series to obtain daily precipitation for 2045 and 2075. For air temperature, the change factors were added to the historical data. Based on median projections for the four major seasons in the region, the calculated change factors for 2045 and 2075 are shown in Table 2.

2.6. Hydrological model setup and input parameters

SWAT was used for the hydrological model. SWAT is a physically based and spatially distributed hydrological model designed for simulating water flow, sediments and contaminant transport within ungauged catchments (Arnold et al., 1998, Arnold et al., 2012). In SWAT, catchments are partitioned into sub-basins and Hydrological Response Units (HRUs), which are lumped land areas within the sub-basin composed of specific land use, soil type topographic characteristics, and management interventions that together describe the spatial heterogeneity of the catchment. For each HRU, key hydrological parameters such as evapotranspiration, runoff, sediment yield and contaminant loads are simulated on daily, monthly or annual time steps. Setting up the model requires various spatial data inputs organized into climate, HRUs, ponds/wetlands, groundwater, and the main channel/reach, which drains the sub-basin. The main input data are Digital Elevation Model (DEM), land use map, soil map and meteorological data. The SWAT simulates the hydrology of a watershed using two stages; the land phase, which controls the amount of water, sediment and contaminant loadings to the main channel, and the routing phase, which describes the movement of the water and associated loads to the channel outlet. This entire hydrologic cycle simulated in SWAT is based on the water balance equation expressed as:

$$SW_t = SW_0 + \sum_{i=1}^{t} (R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw}) \quad (1)$$

where SW_0 and SW_t are the initial and final water content of the soil respectively on day i (mm H_2O), t is the time (days), R_{day} is the amount of input precipitation on day i (mm H_2O), Q_{surf} and Q_{gw} are respectively the amounts of surface runoff and return flow to groundwater on day i, E_a is amount of evapotranspiration, and w_{seep} is the quantity of water that enters the vadose zone from the soil on day i. Thus, the total runoff for day i comprises routed response from each HRU. The model predicts

runoff for each HRU in the watershed, resulting in higher accuracy in estimating the hydrological response of the watershed. In this study, we used the Arc-SWAT version 2012, which is an extension in ArcGIS to obtain the hydrological parameters of the donor catchment (Engesetdalsvatnet Lake). The Engesetdalsvatnet Lake catchment was discretized into 29 HRUs, with 4 slope classes (0–15, 15–30, 30–50, and > 50), and simulated on daily time steps between 2013 and 2017. The Slettebakk and Brusdalen watersheds were respectively divided into 21 and 26 sub-basins, with four slope classes for both. The input meteorological data actually begins from January 2010 to December 2017, and the first three years (2010–2012) was used as a warm – up period.

2.7. Modelling E. coli concentrations in streams

2.7.1. Bacteria wash-off and transport

In the SWAT model, *E. coli* loading within the target catchments were introduced into HRUs in the form of dry manure from the animals within the catchments. Further transport and die-off of *E. coli* in the soil surface layer (top 10 mm), and in the streams were governed by mass transport and die-off/regrowth models (Neitsch et al., 2011). For a given precipitation event in a given day in the catchment, the fraction of *E. coli* (CFU/m^2) washed off from plant foliage onto the soil ($bact_{wsh}$) is calculated using:

$$bact_{wsh} = fr_{wsh} \times bact_{fol} \tag{2}$$

where fr_{wsh} is the wash-off fraction, and $bact_{fol}$ is the amount attached to the foliage. This wash-off fraction is a function of plant morphology, *E. coli* characteristics, and the timing and intensity of the rainfall event. In the model, *E. coli* is assumed to remain in solution in the soil surface layer that interacts with surface runoff. The net change in *E. coli* loading in the surface soil for day i ($\Delta Surface_{Bact}$) can be described as (Niazi et al., 2015):

$$\Delta Surface_{Bact} = Bact_{load} + Bact_{wsh} - Bact_{die} - Bact_{runoff} - Bact_{perc} - Bact_{till} \tag{3}$$

where $Bact_{load}$ is the total *E. coli* loading to the soil on day i, $Bact_{die}$ is the die-off fraction, $Bact_{runoff}$ is the transported fraction through runoff, $Bact_{perc}$ is the concentration percolated into the soil, and $Bact_{till}$ accounts for the concentration percolated into the soil profile through tillage. Similarly, the net change in bacteria in the stream ($\Delta Stream_{Bact}$) is governed by:

$$\Delta Stream_{Bact} = Bact_{inflow} + Bact_{runoff} + Bact_{direct-inputs} + Bact_{resuspension} - Bact_{sed}\,Bact_{die} - Bact_{outflow} \tag{4}$$

where $Bact_{inflow}$ is the initial concentration of *E. coli* on day i, $Bact_{direct-inputs}$ is the *E. coli* concentration from direct deposition from sources such as wildlife, $Bact_{resuspension}$ is the *E. coli* concentration added through resuspension of sediments, $Bact_{sed}$ is the deposited concentration of *E. coli* from sediments, $Bact_{outflow}$ is the *E. coli* transported to lower reaches of the stream, while all the other input parameters are as defined in Eq. (3).

2.7.2. SWAT sediment module

Transport of sediments in SWAT is composed of landscape and channel parts. Daily sediment yield in the landscape component (*Sed*) for day, i (tons) is calculated using the Modified Universal Soil Losses Equation (MUSLE) (Williams, 1975) as:

$$Sed = 11.8 \times (Q_{surf} \times q_{peak} \times Area_{HRU})^{0.56} \times USLE\_K \times USLE\_C \times USLE\_P \times LSUSLE \times CFRG \tag{5}$$

where Q_{surf} is the runoff volume on the soil surface (mm H_2O ha^{-1}), q_{peak} is the peak runoff rate (m^3 s^{-1}), $Area_{HRU}$ is the area of the HRU (ha), *USLE_K*, *USLE_C*, *USLE_P*, and LS_{USLE} are respectively the soil erodability factor (ton m^2 hr)/(m^3-ton cm), cover management factor, support practice factor, and topographic factor for USLE, and *CFRG* is the coarse fragment factor. For the stream channel, the amount of sediment (tons m^{-3}) transported from the reach ($Conc_{Sed,CH,MX}$) is calculated as:

$$Conc_{Sed,CH,MX} = SPCON \times v_{CH,pk}^{SPEXP} \tag{6}$$

where *SPCON* and *SPEXP* are linear coefficient and exponent parameters respectively and are defined by the user. These parameters are used in calculating the channel sediment routing. The parameter $v_{CH,pk}$ is the peak velocity of water in the channel (ms^{-1}).

2.7.3. Bacteria die-off and regrowth

Using first order decay, the fate and transport of *E. coli* from the manure applications, soil and soil water, soil adsorption and runoff partition were modelled using the following equations (Sadeghi & Arnold, 2002):

$$bact_{fol,i} = bact_{fol,i-1} x exp^{(-\mu_{fol,net})} - bact_{min} \tag{7a}$$

$$bact_{sol,i} = bact_{sol,i-1} x exp^{(-\mu_{sol,net})} - bact_{min} \tag{7b}$$

$$bact_{sorb,i} = bact_{sorb,i-1} x exp^{(-\mu_{sorb,net})} - bact_{min} \tag{7c}$$

where $bact_{fol,i}$, $bact_{sol,i}$, and$bact_{sorb,i}$ are the level of *E. coli* in day i, and$bact_{fol,i-1}$, $bact_{sol,i-1}bact_{sorb,i-1}$ are levels of *E. coli* present on foliage, in soil solution, and adsorbed to soil particles on day $i-1$ respectively. μ_{net} is the overall rate constant for die-off/regrowth of *E. coli*, while $bact_{min}$ represents the minimum daily loss of *E. coli*. The effect of temperature on *E. coli* die-off/regrowth in these pools are accounted for by calculating the net rate constants ($\mu_{net,20}$) for each process at 20 °C and adjusted for temperature using:

$$\mu_{net,20} = \mu_{die,20} - \mu_{grw,20} \tag{8}$$

$$\mu_{net} = \mu_{net,20} x \theta_{bact}^{(\bar{T}_{av}-20)} \tag{9}$$

where $\mu_{die,20}$ and $\mu_{grw,20}$ are the rate constants for die-off and regrowth of *E. coli* (day^{-1}) respectively at 20 °C, θ_{bact} is the temperature adjustment factor, and $\bar{T}_{av}$ is the mean daily air temperature. The die-off process in the stream network is modelled as:

$$bact_i = bact_{i-1} x exp^{-WDPRCH} \tag{10}$$

where $bact_i$ and $bact_{i-1}$ are the *E. coli* concentrations in the reach on day i, and day $i-1$ respectively (CFU/100 ml), and *WDPRCH* (day^{-1}) is the die-off rate constant for *E. coli* in the streams.

2.7.4. Calibration and validation of flow and E. coli concentrations in the streams

First, manual calibration of daily flow at the monitoring outlet of the donor catchment was carried out in SWAT from 2013 to 2017, using the preceding three years (2010–2012) as model warmup period. Large number of model parameters present in the SWAT complicates the manual calibration, thus, an automatic calibration and sensitivity analysis was performed to identify the most important parameters that control the discharge level in the catchment. This was achieved using the Sequential Uncertainty Fitting tool (SUFI2) in the SWAT Calibration and Uncertainty Program (SWAT-CUP) (Abbaspour, 2007; Abbaspour et al., 2007). The fitted values of the most sensitive parameters were subsequently applied to the ungauged catchments of the drinking water source (Brusdalsvatnet Lake) to simulate daily flow in the two streams (Slettebakk and Brusdalen) in SWAT from 2013 to 2015. To evaluate the uncertainties and sensitivities of the model parameters in the target catchments, we performed additional calibrations in SWAT-CUP using the flow data transferred from the donor catchment. After the flow simulations, *E. coli* concentrations in the streams were estimated using the bacteria module in SWAT. Validation of the simulated flow in the streams were performed in SWAT-CUP using the fitted parameters and

transferred data for 2016 and 2017 in a single model run. The program was also used to validate the *E. coli* models with the concentrations observed in the streams in 2017, and to determine the parameters that were most sensitive to the concentrations of *E. coli*. Finally, climate scenarios for flow and *E. coli* were simulated for the two streams using the calibrated parameters and the adjusted meteorological data.

2.7.5. Model uncertainty and sensitivity analysis

Uncertainties in the estimated flow and *E. coli* concentrations in the streams were evaluated using the SUFI2 procedure in the SWAT-CUP program. Using initial ranges of parameters taken, the SUFI2 procedure caries out a number of specified iterations until the 95% prediction uncertainty (95PPU) and the average distance between the 2.5th and 97.5th percentiles is smaller than the standard deviation of the measured data. The parameter combinations within the parameter ranges are used to find the best parameter ranges that yield the "best-fit" (Abbaspour, 2007). The algorithm also calculates two statistics; P-factor, which indicates the percentage of the observed data enveloped by the model results, and the R-factor, which shows the thickness of the 95PPU envelop. These statistics are used to quantify the fit between simulation results, and generally, larger values considered more efficient (Abbaspour, 2007). Other hydrological model efficiency indices such as the Nash-Sutcliff efficiency (NS), and the coefficient of determination (R^2) are calculated by the algorithm. In this study, the NS objective function was used as the objective function for comparing the simulated and observed daily flows. Further, the algorithm determines parameter sensitivity by regressing the Latin hypercube generated parameters against the values of a chosen objective function. A *t – test* is then used to evaluate the relative significance of each parameter (more sensitive parameters have larger absolute values of the t- statistic). In this study, the sensitivity and uncertainty analysis for flow and *E. coli* concentrations each involved one-at-a-time option for visual inspection of the effect of each parameter, and the global sensitivity analysis for ranking the parameters based on their t-statistics and p-values. Based on the t-statistics, we ranked the parameters from 1 (most sensitive) to *n* (least sensitive).

3. Results and discussion

3.1. Stream flow and parametrization

The ranges, fitted values, and the sensitivity ranks of the 25 hydrological parameters calibrated in the donor and target catchments are summarized in Table 3. The difference between some of the parameters in the two catchments (donor and target) may reflect disparities in watershed characteristics such as land use and altitude that are local to each catchment. Comparison of the parameters in the target catchment with those obtained in the calibration of the donor catchment revealed no distinct disparities in the sensitivities. This indicates that both catchments have similar hydrological responses. The results illustrates that the canopy storage (CANMX) and plant uptake compensation factor (EPCO) were the most sensitive parameters to runoff generation in the catchments of the streams, with t-statistics of 1.22 and 1.21 respectively. The other parameters that showed high sensitivities were; curve number (CN2), effective hydraulic conductivity in the main channel (CH_k2), soil organic carbon content (SOL_CBN), groundwater delay time (GW_DELAY), saturated hydraulic conductivity of the soil (SOL_K), and snowmelt base temperature (SMTMP), which showed smaller t-statistics (between 0.05 and 0.77).

All other parameters had negative t-statistic values of up to −1.8. While CANMX and EPCO were most sensitive to the flow in this study, the two parameters have not often been rated as very essential in other studies. For instance, in the study of Kim et al. (2010), these parameters were ranked 11 and 15 respectively. In another study that used SWAT to simulate the fate and transport of *E. coli* in a river, the EPCO was not reported among the top six parameters sensitive to flow in the river (Niazi et al, 2015). These studies were conducted in areas with significant forest cover, just as the location of the watershed in this study. The high sensitivities of these parameters in this study may be due to interception of precipitation by the dense forest canopy (constituting 67% and 63% of land cover in the donor and target catchments respectively), as well as the morphology of the plant species present in this study area. The forest in the catchments comprise temperate coniferous forest pine, with low fractions of broadleaf evergreen and deciduous trees and grasslands. Moreover, during the manual calibration of the flow model in the donor catchment, the EPCO parameter was increased to allow higher compensation for plant uptake, since lower flow levels were generated at lower EPCO values. Increasing the value of this parameter enables deeper soil layers to contribute towards compensating for water used by plants (Neitsch et al., 2011), thereby increasing the generated flow.

Other parameters such as the curve number CN2, which is controlled by the antecedent moisture condition of the soil, was ranked 3rd in this study. This parameter is used to calculate the runoff, and greatly depends on the soil characteristics. The ranking of this parameter in this study is consistent with results of other studies in which the parameter is often ranked as one of the most important parameters in flow calibration with SWAT (Kim et al., 2010; Niazi et al., 2015; Kim et al., 2017). In addition, hydraulic conductivity in the main channel (CH_K2), which ranked as the 4th in this study has been rated high in many studies that calibrated flow in tropical areas (Vilaysane et al., 2015; Rafiei Emam et al., 2016; Kim et al., 2017). The content of organic matter in the soil was very sensitive to the generated flows in both the donor and target catchments, ranking 5th among the flow parameters. As indicated in Section 2.3 of this paper, humus and peat constitute 53% and 63% of the two catchments respectively. The properties of peat cover and vegetation, as well as the pattern and orientation of peatland surface are among the major factors that regulate catchment runoff in forested peatlands (Emili and Price, 2006; Kværner and Kløve, 2008). Watersheds dominated by peat soils often have high infiltration rates, and are able to hold water for a long time, resulting in low runoff potential. When they become fully saturated however, intermittent runoff can be generated due to their shallow water table (Menberu et al., 2014). Although snowmelt is an important factor in overland flow generation in this region, only the base rate temperature of snowmelt was ranked among the top ten most sensitive parameters in this study. For the four other snowmelt parameters, the snow pack temperature lag factor (TIMP), showed highest sensitivity, ranking 14th. These snow parameters (SMTMP and TIMP) were ranked among the top ten most sensitive parameters in a related study (Kim et al., 2010).

Results of the estimated streamflow hydrographs between 2013 and 2015 (calibration period) and from 2016 to 2017 (validation period) are shown in Fig. 2. The calibration output generally showed very good agreement with the transferred flows in both streams. Peak flow periods were not simulated with high precision. In addition, the model underestimated low flows occurring in the summer, particularly in the Slettebakk stream (Fig. 2 a). This may be caused by factors such as deficiencies in the climate data, the flow regionalization process, and characteristics specific to the donor and target catchment. As indicated in Section 2.1 of this study, both the donor and target catchment receives high amount of rainfall throughout the seasons, resulting in significantly high flows in the streams throughout the year. For the average of 1987 mm of rainfall received annually in the main catchment of the Brusdalsvatnet Lake (which includes the sub-catchments of the modeled streams), rainfall in summer (June – August) and winter (December – February) in this catchment are 688 mm and 1298 mm respectively. However, peak flows are often higher in the spring (March – May) and autumn (September – November).

The performance indices of the flow and bacteria models as well as related results are shown in Table 4. The value of the Nash-Sutcliffe efficiency (NS) for the flow calibration of the Slettebakk and Brusdalen

Table 3
Water Flow model parameters and their sensitivity ranks for the donor catchment (1 for most sensitive to 36 for less sensitive).

Parameter	Description	Range	Fitted		Sensitivity
			Donor	Target	
CN2.mgt	Curve number (–)	−0.02–0.20	0.02	0.02	3
Alpha_BF.gw	Base flow recession constant (–)	0–1	0.52	0.45	24
GW_DELAY.gw	Groundwater delay time (days)	30–450	98	93	6
GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow (mm)	0–2	0.13	0.1	15
GW_REVAP.gw	Coefficient for water movement from shallow aquifer into the unsaturated zone (–)	0.02–0.2	0.12	0.11	13
REVAPMN.gw	Threshold depth of water in the shallow aquifer required for bottom – up water movements (mm)	0–1000	861	850	16
RCHRG_DP.gw	Deep aquifer percolation fraction (–)	0 – 1	0.82	0.85	19
ESCO.hru	Soil evaporation compensation factor (–)	0 – 1	0.92	0.95	20
EPCO.hru	Plant uptake compensation factor (–)	0–1	0.94	0.95	2
CANMX.hru	Canopy storage (mm)	0–100	41	45	1
OV_N.hru	Manning's roughness coefficient for overland flow ($m^{1/3}s^{-1}$)	0–100	43	45	12
LAT_TIME.hru	Lateral flow travel time (days)	0–180	23	27	17
SURLAG.bsn	Surface runoff lag coefficient (–)	1–24	14.6	15.9	10
CH_K1.sub	Manning's roughness coefficient for tributary channels flow ($m^{1/3}s^{-1}$)	0–300	165	165	9
CH_N1.sub	Effective hydraulic conductivity in tributary channels alluvium (mm h^{-1})	0.01–30	22	22.5	11
CH_N2.rte	Manning's roughness coefficient for main channel flow ($m^{1/3}s^{-1}$)	0.01–0.3	0.28	0.28	21
CH_K2.rte	Effective hydraulic conductivity in main channel alluvium (mm h^{-1})	0–500	25	25	4
SOL_AWC.sol	Available soil water capacity (mm H_2O/mm $soil^{-1}$)	0–1	0.16	0.15	23
SOL_CBN.sol	Soil organic carbon content (%)	0.05–10	2.58	2.54	5
SOL_K.sol	Saturated hydraulic conductivity (mm h^{-1})	0–2000	103	100	7
SFTMP.sno	Snowfall temperature (°C)	−5–5	−0.7	−0.5	25
SMTMP.sno	Snow melt base temperature (°C)	−5–5	−2.3	−2.5	8
SMFMX.sno	Maximum melt rate for snow during year (mm H_2O/°C/day)	0–10	9.5	9.5	22
SMFMN.sno	Minimum melt rate for snow during year (mm H_2O/°C/day)	0–10	1.5	1.5	18
TIMP.sno	Snow pack temperature lag factor	0–1	0.97	0.94	14

streams were respectively 0.68 and 0.70, while the coefficient of determination (R^2) were 0.72 and 0.74 respectively. In addition, values of p-factor and r-factor, which respectively indicate the percentage of the transferred observations included within the 95% prediction uncertainty band (99PPU), and the thickness of the uncertainty band (Abbaspour et al., 2015) are also presented in Table 4.

3.2. *E. coli* concentration and parameterization

Table 5 shows the results of the 15 calibrated *E. coli* model parameters, which include sediment transport related parameters and their sensitivities. The two parameters most sensitive to the temporal variation of *E. coli* in the streams were the exponent parameter for calculating sediment in channel routing (SPEXP), and the peak rate adjustment factor for sediment routing in the sub-basin (ADJ_PKR). The calibrated values for these parameters were respectively 1.33 and 1.32 respectively. *E. coli* variation was also sensitive to other sediment related parameters such as the linear parameter (SPCON) and the channel bank erodability factors (CH_COV1 and CH_COV2). These parameters mostly control the amount of sediments in the stream channels. Their high sensitivities to the concentrations of *E. coli* in the streams may be due to the roles of sediment resuspension, transport deposition processes in the streams that provide protection and nourishment for the *E. coli*. Sediments can serve as either sources or sinks of *E. coli* concentrations in freshwater (Ishii et al., 2007), provide soluble organic matter and nutrients for the survival of bacteria, and shield them from predators and exposure to UV radiations in sunlight (Jamieson et al., 2004; Sinclair et al., 2012). Furthermore, attachment to streambed sediments is one of the most important mechanisms that increases the persistence of bacteria in water. Therefore, bacteria concentrations in water can increase during high flow events due to resuspension of bottom sediments. Results of a study that used tracer bacteria to investigate the resuspension and persistence of sediment-related bacteria in a stream showed that enteric bacteria can survive in bed sediments for up to six weeks (Jamieson et al., 2005). Moreover, the attachment ratios of *E. coli* to streambed sediments have been shown to increase with decreasing sediment sizes, and the attachment and resuspension rates of the bacteria varied with different types of sediments studied (Cervantes, 2012). Further, streambed sediment resuspension has been demonstrated to increase *E. coli* concentrations in streams by up to two orders of magnitude (Muirhead et al., 2004). Other parameters that showed high sensitivities were the decay factor of *E. coli* in the stream (WDPRCH), bacteria partition coefficient (BACTKDQ), bacteria partition coefficient (BACTKDDB), and the parameter that indicates the fraction of animal manure that contains active colonies of the bacteria (BACTKDDB). These parameters mainly control availability of bacteria on the soil surface to be transported by runoff. The high sensitivities of the sediment related and the other deposition related parameters found in this study are consistent with results of other studies in which these parameters were ranked among the top ten (Niazi et al., 2015; Kim et al., 2017).

Fig. 3 shows comparisons of the concentrations of *E. coli* from the model calibrations and the biweekly observations in the streams. *E. coli* data for January – February and the entire summer period (June – August) were not available. The performance statistics for the models were not very good (NSE = 0.30, R^2 = 0.31 for Slettebakk stream, and NSE = 0.24, R^2 = 0.26 for Brusdalen stream). The low performances of the models can be partly attributed to uncertainties in concentrations of *E. coli* inputs from the farm and wild animals in the catchments. As indicated in Section 2.4 of the paper, 14 observations were used in the *E. coli* model calibrations. It is worth noting that the bacteria module in SWAT only simulates the *E. coli* concentrations based on average concentrations in animal manure. The calibrations are separately performed to compare the model outputs with the observations such that performance indices can be calculated. Thus, the model performance indices could have been higher if the calibrations were performed with larger number of observations. Despite the low accuracies, the trends of the model predictions generally agreed with the pattern of observations of *E. coli* in both streams. The results indicate that lower concentrations of *E. coli* in the streams occur in the winter months (December - February). The concentrations become higher in the latter part of the spring season (May), and begin to decrease from the onset of the autumn season in September. Although the catchments receive relatively low precipitation in the summer months with low flows in the streams (Fig. 2), the *E. coli* concentrations in the streams are highest in this season. This could be as a result of higher availability of wildlife in the

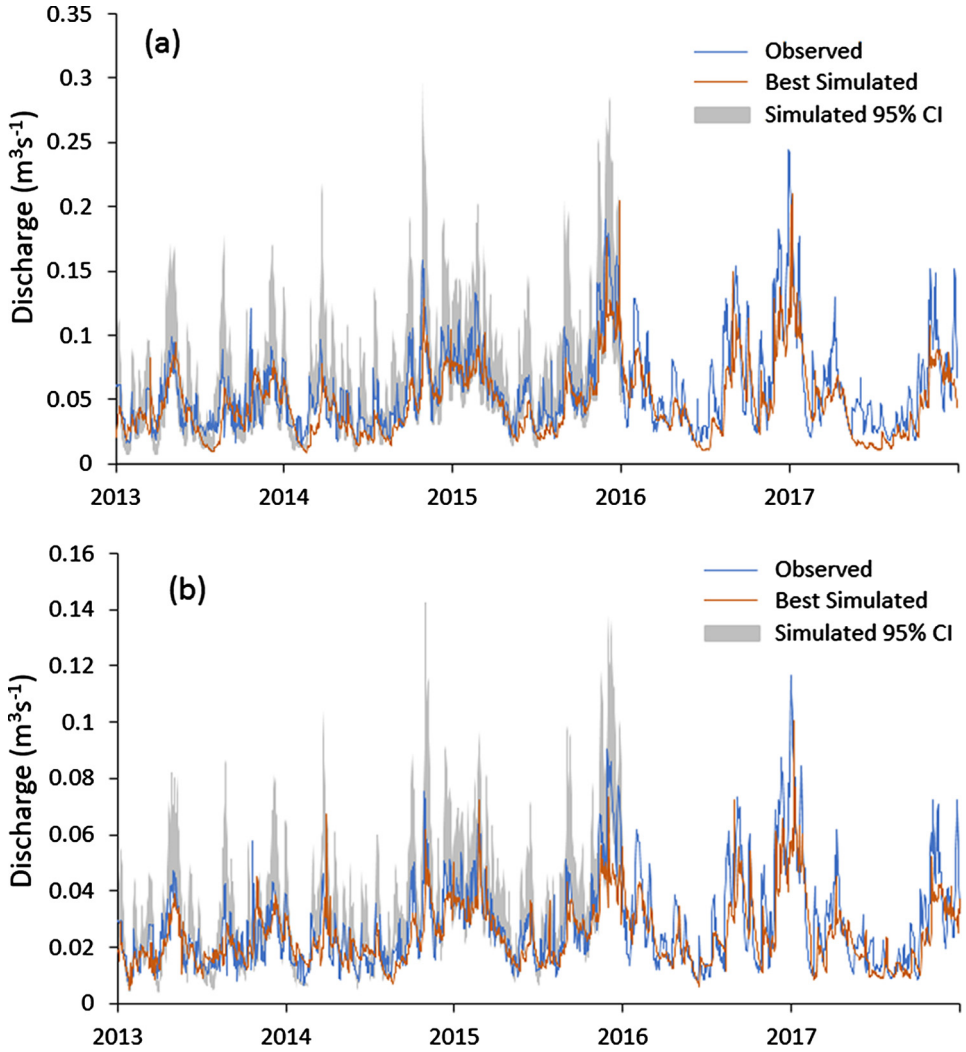

Fig. 2. Calibration (2013–2015) and validation (2016–2017) of daily flow in the Slettebakk stream (a) and Brusdalen stream (b) compared with regionalized discharge from Engesetdalsvatnet Lake (donor catchment).

Table 4
Model performance indices for water flow and *E. coli* concentration in the Slettebakk and Brusdalen streams.

Model	NS	R^2	p-factor	r-factor
Discharge				
Slettebakk stream	0.68	0.72	0.72	0.63
Brusdalen stream	0.70	0.74	0.71	0.63
E. coli				
Slettebakk stream	0.30	0.31	0.28	0.16
Brusdalen stream	0.24	0.26	0.22	0.11

catchments during summer, which is warmer than the other seasons. Although the main sources of *E. coli* contamination in the Brusdalen stream (Fig. 3b) was assumed to be horses and sheep raised in the area, 84% of its catchment is covered by forest, while the Slettebakk catchment is composed of 94% forest. Thus, the occurrence of higher *E. coli* concentrations in both streams in summer may be an indication that wildlife are important sources within both catchments. Moreover, while precipitation plays a major role in determining the concentration of FIB in surface water, it could also cause dilution of concentrations of contaminants in the stream due to high flows and faster moving stream water. Low flows in summer could however result in the microorganism travelling a shorter distance and longer settling time in the streams, as these factors also affect pathogen concentrations in surface water (Schijven et al., 2013). Although high temperature in summer can increase the die-off rates of *E. coli* in the streams, high temperature could also enhance the persistence of the microorganism in farm animal faeces within the catchments (Oliver and Page, 2016), potentially leading to high concentrations in the streams. Further, lower *E. coli* concentrations mostly in the winter and spring seasons may be caused by snow cover on soil surface and in the streams, which can reduce the fraction of animal manure washed into the streams.

3.3. Results of climate scenarios

Figs. 4 and 5 show the results of the climate scenarios for flow and *E. coli* concentrations in the streams. In the flow projections (Figs. 4a and 5a), average daily flows were calculated from the observed values and the simulated climate data for the five-year period. The results indicate that flows in both streams may substantially increase in the future. Moreover, pattern of the flows in the future can be different from current patterns. For instance, while the flows in the streams begin to rise at the onset of the autumn season (September) in the historical data, this event may shift backwards, since the flows start to rise in July for both 2045 and 2075. Thus, the current lower flow period in summer will be shortened. Further, peak flows in the spring season may occur earlier (winter) in the future. From the calculated change factors for the climate variables (Table 2), differences in median expected change in precipitation in the region across the various seasons for 2045 and 2075

Table 5
E. coli model parameters for the Slettebakk and Brusdalen streams and their sensitivity ranking.

Parameter	Description	Range	Fitted	Sensitivity
BACT_SWF.bsn	Fraction of manure applied to land areas with active colony forming units (–)	0–1	0.25	9
WOF_P.bsn	Wash-off fraction for persistent bacteria (–)	0–1	0.95	15
BACTKDQ.bsn	Bacteria partition coefficient (–)	0–500	225	7
BACTKDDB	Bacteria partition coefficient in manure (–)	0–1	0.63	8
BACTMX.bsn	Bacteria percolation coefficient (–)	7–20	7.65	11
WDPRCH.bsn	Die-off factor for persistent bacteria in streams at 20 °C (–)	20–80	59	4
ADJ_PKR.bsn	Peak rate adjustment factor for sediment routing in the sub-basin	0.5–2	1.32	2
SPCON.bsn	Linear parameter for calculating the maximum amount of sediment that can be re-entrained during channel sediment routing (–)	0.0001–0.1	0.007	3
SPEXP.bsn	Exponent parameter for calculating sediment re-entrained in channel sediment routing (–)	1–1.5	1.33	1
CH_COV1.rte	Channel bank erodability factor (–)	1–5	2	6
CH_COV2.rte	Channel bed erodability factor (–)	1–5	3.2	5
FILTERW.mgt	Width of edge-of-field filter strip (m)	0–100	95	13
BIOMX.mgt	Biological mixing efficiency (–)	0–1	0.65	12
USLE_P.mgt	USLE equation support practice factor (–)	0–1	0.65	10
USLE_K.sol	USLE equation soil erodability factor (–)	0–1	0.05	14

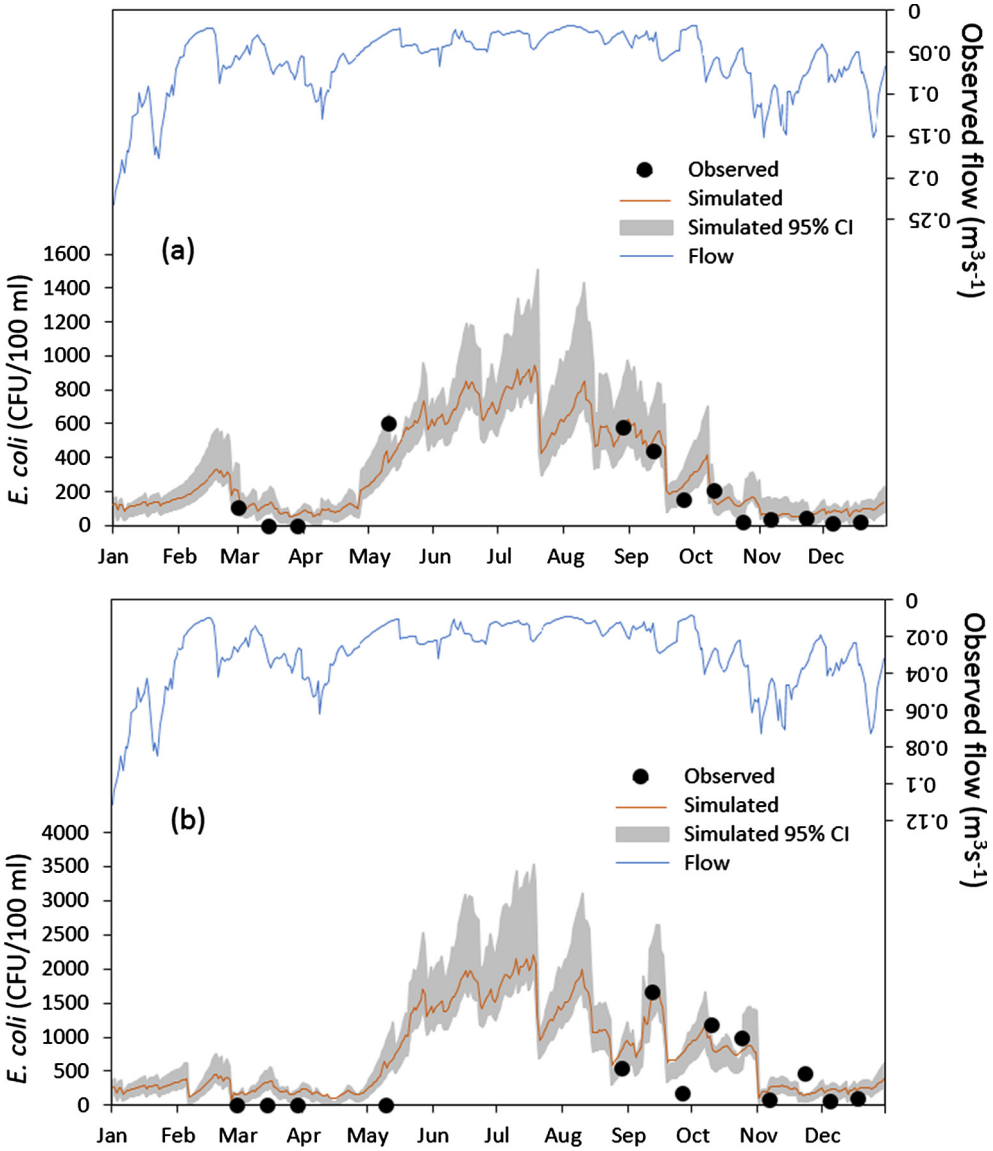

Fig. 3. Temporal variation of observed *E. coli* concentrations in the Slettebakk stream (a) and Brusdalen stream (b) in 2017 compared with calibrated model results for 2017.

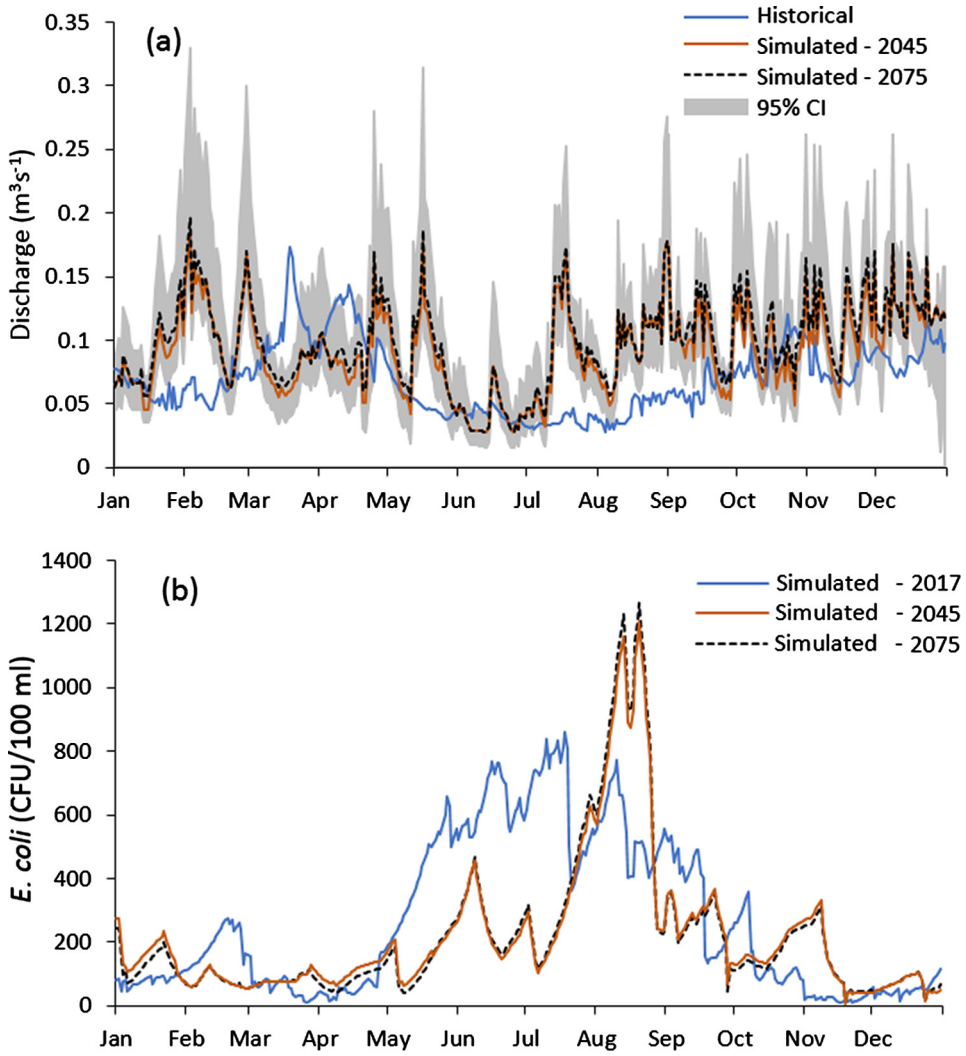

Fig. 4. Comparison of (a) average historical flow (2013–2017) and simulations for 2045, 2075 in Slettebakk; (b) Simulated *E. coli* in the stream for 2017, 2045, and 2075.

is very small. The calculated annual projected changes are 1.03% and 1.05% respectively. Accordingly, the simulated flows in these years do not significantly vary, although marginal increases may occur in the spring and autumn seasons in 2075 compared to the 2045 levels.

Potential variations in the concentrations of *E. coli* in the Slettebakk and Brusdalen streams in 2045 and 2075 (Figs. 4b and 5b respectively) indicate similar patterns of variation. The results suggest that under the current climate projections for the region, concentrations of *E. coli* in the streams in 2045 and 2075 may remain similar to current levels observed in the winter and spring seasons. Possible future variations in the numbers of wild/farm animals, as well as other factors including future land use changes within the catchments can affect the predicted results. However, due to uncertainties associated with these factors, it was not possible to account for them in the model. Compared to the high levels presently occurring throughout the summer, lower concentrations might be observed in both streams in the future during the first two months of summer (June and July). However, the autumn turnover period (late August to early September) may introduce high concentrations of *E. coli* into the streams. The potential variation in the patterns of discharge into the streams in the future as shown by the model predictions suggest that flow into and out of the drinking water source (Brusdalsvatnet Lake) could vary in future. Although the Lake has high residence time of approximately 8 years, changes in the total inflows from the over 150 tributaries may considerably influence circulation patterns in the Lake. This could lead to the occurrence of higher temperatures and higher concentrations of FIBs and other contaminants at the raw water intake point of the Ålesund water utility in late summer. These potential shifts in the periods for which higher concentrations of contaminants are known to occur historically in the Lake would provide the water utility managers the necessary information for developing climate adaptation strategies.

Potential sources of uncertainties in the results of this study include the historical input data used for calibrating the models, regionalization of measured flow, the projected climate models used to adjust the meteorological variables, model structure and parameters. However, due to lack of observations for future meteorological variables, such uncertainties cannot be directly assessed, and therefore difficult to reduce (Beven, 2016; Meresa et al, 2017). Moreover, understanding of the hydrological processes and the essential parameters affecting runoff generation and microbial discharge into drinking water sources in Norway is necessary for proactive decision making that will augment long-term climate adaptation strategies for water utilities.

4. Conclusion

We implemented a hydrological model to assess the impact of local climate projections on flow regimes and *E. coli* discharge in the ungauged catchment of the drinking water source for the city of Ålesund

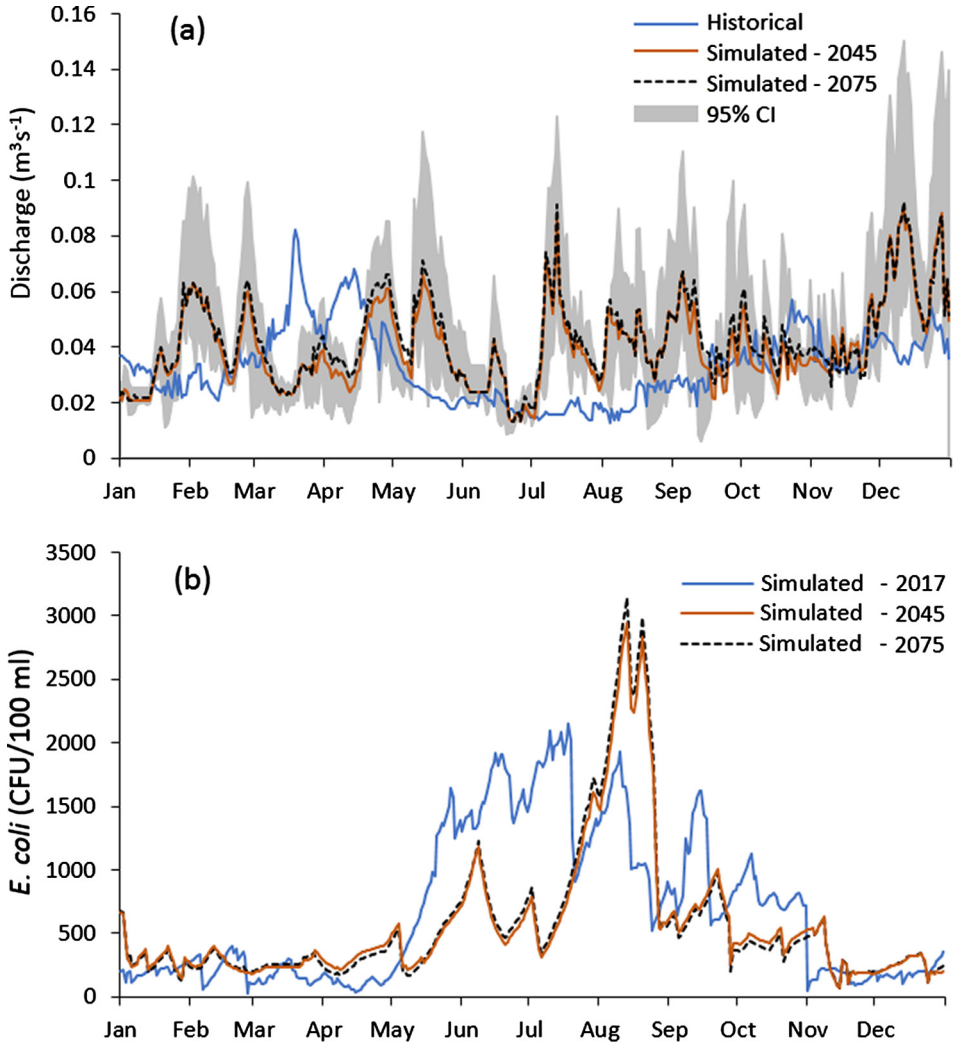

Fig. 5. Comparison of (a) average historical flow (2013–2017) and simulations for 2045, 2075 in Brusdalen; (b) Simulated *E. coli* in the stream for 2017, 2045, and 2075.

in Norway. The results illustrates that regionalization method is a highly efficient method for predicting discharge in ungauged catchments. Results of the climate scenarios for the flow indicate that under the current climate projections for Møre and Romsdal county of Norway, peak flows in the streams, which currently occur in spring (March – May) and autumn (September – November) may be shifted back to winter (December – February) and summer (June – August) respectively. The *E. coli* concentrations in the streams could however remain the same in winter and spring, while the high concentrations currently observed in summer may be low in future. However, compared to current levels, *E. coli* concentrations may be higher at the start of autumn in future (late August – early September).

With variations in the discharge patterns expected in the future, circulation periods in the Lake resulting from discharges from its tributaries may be shifted back from spring and autumn to winter and summer respectively. This implies that high concentrations of *E. coli* and other potential contaminants may occur at the raw water intake earlier (late summer) than the current period (in autumn). High *E. coli* concentrations in rivers and streams indicate the potential presence of harmful pathogens that pose a public health risk to the people through either recreational activities or consumption of water produced from such waterbodies. The health risks associated with water consumption from these rivers and streams can be significantly reduced if the expected time of their occurrence in the future are known by water utility managers. This would enable the planning and implementation of optimal water treatment processes that accommodate the potential changes. Results of this study have implications for management decisions in watersheds of drinking water sources in Norway, where over 90% of water utilities rely on surface waterbodies.

Declaration of interests

None.

Acknowledgements

The authors of this work are grateful to the management of the water utility in Ålesund, Norway, the Norwegian Water Resources and Energy Directorate (NVE), the Norwegian Institute for Bioeconomy (NIBIO), and the Norwegian Geological Survey (NGU) for the data used in this study.

Funding

This work is part of the KLIMAFORSK project – On the Association between climate change and Waterborne Diseases funded by Research Council of Norway (Project No: 244147/E10).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2019.04.021.

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